Handbook of Ecological Restoration

Volume 2
Restoration in Practice

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INTRODUCTION

Seagrasses are marine flowering plants consisting of 12 genera and approximately 60 species growing in all of the world's oceans with the exception of the most polar regions (den Hartog, 1970; Phillips & Meñez, 1988). Nearly all seagrasses grow in unconsolidated sediments in water depths ranging from the intertidal zone to as deep as 35–50 m. They are vascular plants anchored to soft sediments by a functional and complex rhizome and root system, with the exception of the genus *Phyllospadix* which grows on solid substrates along the Pacific coast of the United States. The seagrass leaf canopy baffles the flow of water, and together with their rhizome and root mat seagrasses stabilise sediments, cleanse the water column of fine particles, and recycle nutrients between the sediments and overlying waters (Fonseca, 1996). Numerous species of invertebrates and large vertebrates consume seagrasses as a portion of their diet, and the complex structure and physical stability provided by seagrasses form the basis for productive ecosystems consisting of plant and animal epiphytes, benthic macroalgae, invertebrates, mobile vertebrates and numerous other organisms (Thayer *et al.*, 1984). Many of the animal species that utilise seagrasses rely on their structural complexity to provide shelter and sources of food for their juvenile stages. This is one of the most important biological functions of the seagrass ecosystem.

The lack of taxonomic biodiversity in seagrasses is compensated by a wide diversity of size and morphological growth forms. The size and biomass of seagrass varies over an order of magnitude (Kenworthy *et al.*, 2000), resulting partly from genotypic differences as well as from phenotypic plasticity within individual species. For example, the canopy height of the smallest species known, *Halophila decipiens*, usually never exceeds 10 cm while species of *Zostera* can have canopies exceeding 5–7 m in height. The diversity of clonal growth forms and sexual reproductive strategies is accompanied by phenotypic variation that allows the limited number of seagrass species to occupy a wide range of environmental conditions from wave-swept shorelines to relatively deeper regions of continental shelves. Only a few other macroscopic plants growing in the ocean are capable of filling the niche type that seagrasses occupy.

Even though taxonomic biodiversity is limited in seagrasses, the diversity of size and morphological forms is accompanied by different growth and survival strategies uniquely adapted to the environments where the plants thrive. The range of growth strategies is also responsible for the patterns of seagrass bed development seen throughout the world. This is especially evident in multi-species tropical seagrass communities where distinctive successional processes are evident in the formation of stable climax communities and in their response to disturbance (Zieman, 1982). In tropical seagrass communities, colonising and climax species can be readily distinguished from one another and the unique attributes of these species can be utilised to enhance their protection and restoration (Fonseca *et al.*, 1987).

RATIONALE FOR RESTORATION

Fortunately, in many countries, the battle to recognise seagrasses as critical coastal ecosystems worthy of conservation and restoration has been won.

This recognition can be credited to the publication of thousands of papers from dozens of countries around the world representing years of research. To the best of our knowledge, research has yet to record a seagrass bed which is anything but a faunal-rich, highly productive ecosystem, that stabilises the sea floor, limits coastal erosion and filters the water column (Wood *et al.*, 1969). Thus, the ecological and sociological value of seagrasses has been broadly established (Wyllie-Echeverria *et al.*, 2000). Where these values are not recognised, it often appears to be the result of local political and development interests overriding conservation values (personal observation).

Threats to seagrass ecosystems and causes of degradation arise from a wide variety of sources. Eutrophication, coastal construction, motor vessel operation, fishing practices and many other activities have led to both local and regional losses of seagrasses (Short & Wyllie-Echeverria, 1996; Fonseca *et al.*, 1998*a*). Losses of seagrass also occur through natural processes such as disease (Muelstein, 1989; Robblee *et al.*, 1991), tropical cyclones (Preen *et al.*, 1995) and overgrazing by invertebrates (Rose *et al.*, 1999). Where the species composition and life-history strategies promote recolonisation, seagrasses can recover naturally from perturbations (Preen *et al.*, 1995). However, in many instances either the severity of the environmental modification responsible for the declines or the extremely slow rate of natural recovery leads to long-term losses. For example, in climax tropical communities dominated by *Thalassia testudinum* the time to full recovery in severely damaged vessel grounding sites can be more than a decade (Kenworthy *et al.*, 2000; Whitfield *et al.*, in press). In these instances, loss of seagrasses leads to numerous undesirable and difficult-to-reverse conditions, most importantly the elimination of habitat structure and the sediment stabilisation properties of the canopy and rhizome mat. A negative feedback on the ecosystem results; once the seagrass cover is lost and with it the self-sustaining properties of the system provided by the seagrasses, modification of the sediments and degradation of the water column may proceed without interruption. Seagrass restoration then becomes a much more difficult task, because it is nearly impossible to replace the attributes seagrasses provide, and a way to correct the physicochemical properties of the system must be found before reintroduction of the seagrasses can begin.

We posit that the issues regarding seagrass restoration are not the technology of planting and raising seagrass beds, but the failure to apply basic ecological principles in implementing restoration actions. Seagrasses can be readily transplanted and when sites are appropriately selected (see below and discussion in Fonseca *et al.*, 1998*a*), significant restoration successes have emerged. In fact, new technologies are continually being developed in both the deepwater (Perth, Western Australia: Fonseca *et al.*, 1998*b*; E. Paling, personal communication) and shallow water (Tampa Bay, Florida: J. Anderson, personal communication) approaches. Also, improvements in large-scale seeding techniques are being advanced which have promise with some seagrass species (Granger *et al.*, 2000; Orth *et al.*, 2000). We are only just beginning to recognise the many situations in which opportunities for substantial restoration have either been squandered or serious mistakes in site selection have been made, largely because those involved did not understand the habitat requirements and/or the life history of the plants with which they were working.

The ecological value of seagrasses translates into enormous commercial and social benefits. For example, in the Indian River Lagoon, Florida seagrass meadows have been described as the marine equivalent of tropical rainforests providing the ecological basis for fisheries worth about US$25 000 per hectare or a total of approximately 1 billion dollars a year (Virnstein & Morris, 1996). Seagrass-dependent fisheries and wildlife communities are the economic foundation for commercial and recreational fishermen as well as for a variety of industries and people that utilise the coastal zone for commerce and personal enjoyment. Socially, these values are transferred to the health and well-being of the families of these user groups and the regional economies of nations worldwide. The many physical, biological, economic and social attributes combine to make seagrasses an essential and ecologically important habitat in coastal marine ecosystems (Wyllie-Echeverria *et al.*, 2000); consequently they are in need of restoration where they have been anthropogenically injured or lost (Fonseca *et al.*, 1996; Sheridan, 1999).

PRINCIPLES OF RESTORATION

We base our assessment of the status of seagrass restoration on a perspective from within the United States legal framework. Seagrass beds in United States coastal waters are generally viewed as public trust resources, and such injuries to these resources are considered losses suffered by the public. A number of federal and state laws include liability provisions which allow the public to be compensated for injuries to seagrasses (for example, the United States's National Marine Sanctuary Act of 1972, 16 USC 1431 *et seq.*). To evaluate this loss in a fair and reasonable manner, we must consider not only the static loss in area and/or degree of the injury, but also the loss of resource services provided by the seagrass bed between the time it is injured and the time it recovers to 100% of pre-injury conditions (Fonseca *et al.*, 2000*a*). This approach is consistent with the 'no-net-loss' of wetlands policy that has become a benchmark of restoration strategies in the United States. Our more recent approach substitutes for the 'mitigation' or 'replacement ratio' used to identify the amount of habitat to be generated to offset the amount lost. In the past, use of replacement ratios has frequently led to undercompensation of lost resources because lost interim ecological services were not addressed.

Effecting no-net-loss and achieving recovery of interim resource services requires that the injured site be fully rehabilitated (on-site restoration), alternative compensatory restoration sites be found (off-site) or some degree of both. To limit the scope of discussion, we are focusing on in-kind restoration (i.e. seagrass service loss replaced by seagrass service gains). On-site restoration can often be achieved, but may require engineering interventions to 'fix' the site, such as filling excavation holes caused by vessel groundings or altering water flow. Off-site selections, in our experience, have had higher probabilities of restoration failure because inexperienced resource managers choose inappropriate sites. They are frequently under the impression that open habitat areas are prime sites for restoring seagrass when, in reality, the sites selected either cannot support seagrass, or currently support only low levels of seagrass. This fallacy has been addressed in detail in several publications (Fredette *et al.*, 1985; Fonseca *et al.*, 1987, 1998*a*; Fonseca, 1992, 1994). Suffice it to say, Fredette *et al.*'s (1985) condition 'If seagrass does not grow there now, what makes you think it can be established?' best sums the problem. Recently, Calumpong *et al.* (in press) listed the criteria for off-site selection that can be used to avoid off-site selection problems. By giving attention to these details of site selection, the probability of successful restoration can be greatly enhanced.

RESTORATION IN PRACTICE

We have dealt previously with what we consider to be the status of this aspect of restoration (Fonseca *et al.*, 1998*a*). However, there are at least four major deficiencies in the process of seagrass restoration. First, the choice of an appropriate metric for evaluating restoration has been elusive. We present here for the first time findings from a panel of United States seagrass experts that considered what are the appropriate metrics for tracking the performance of a seagrass restoration project. Second, setting fair, reasonable and consistent ratios for replacement of damaged seagrasses has also been at issue. We review the methodology used by the National Oceanic and Atmospheric Administration (NOAA) for defining the interim loss of resource services accrued by damage to seagrass beds and the process of computing compensatory restoration. Third, we feel that the weakest part of seagrass restoration has been the selection of the restoration site. We delve into the pitfalls of site-selection strategy – the point in the process where most plans go awry. For completeness, we briefly review the extant methodologies for restoring seagrass beds. Fourth and finally, finding a realistic basis for computing cost of these projects has been a vexing issue for years. Here we provide an evaluation of cost for the planning, implementation and monitoring of a seagrass restoration project based on a United States federal court case successfully prosecuted by NOAA.

Definition of injury and evaluation of lost interim services

Defining lost resource services

Computation of lost resource services requires three assessments: (1) area of habitat lost; (2) the length of time needed for the functions associated with that area (and lost to the ecosystem at large during the period of the injury) to recover to their pre-impact levels; and (3) the shape of that recovery function (Fonseca *et al.*, 2000*a*). Using seagrass ecosystems as an example, if 1 hectare of seagrass were destroyed today and replanted tomorrow and, for argument's sake, reached standards of equivalency (e.g. shoot density, biomass, coverage) in two years, the interim loss of ecological services over this two-year period would be relatively low. However, if the restoration of this site were not undertaken immediately and if the site required seven years to reach its pre-impact state, the level of compensation due the public for the interim losses from this same 1-hectare injury would be substantially higher. This highlights the weakness of fixed compensation ratios.

Actual projects rarely enjoy tight temporal coupling between either the injury and on-site repair work, or between the injury and the additional restoration required to compensate for the ecological services lost from the time of the injury until full recovery. Among other issues, it is very difficult to consistently locate and successfully create new seagrass habitat that meets ecologically responsible site-selection criteria, especially those criteria which preclude simply substituting naturally unvegetated bottom for vegetated bottom (Fonseca *et al.*, 1998*a*). Finding large areas of suitable substrate for restoration in close proximity to the impacted area is rare, and often results in restoration at sites physically removed from the impact area. Thus, any functions affected by spatial elements of ecosystem linkages are lost (i.e. geographic setting). Second, the lost production was removed from a specific point in time. Therefore, in some instances it cannot be returned in a way to avoid disruption of ecosystem functions, such as the loss of last year's spawn of herring or set of bay scallops that might occur as a result of injury to a seagrass bed. Moreover, if there were a longer period of time between the injury and full recovery from the injury, then one could argue that replanting conducted a long time after an impact has less value than ones conducted sooner. This realisation is the basis for NOAA's more recent approach to objectively and quantitatively standardise the problem of computing interim lost services by habitat equivalency analysis (HEA). This approach provides a basis for setting replacement ratios and arriving at a quantity of persistent area of given quality that has been defined as an appropriate metric of success (Fonseca, 1989, 1992, 1994; Fonseca *et al.*, 1998*a*, 2000*a*).

Determination of interim loss and its implementation into the restoration process is tightly integrated with the establishment of a restoration plan. While such a plan must identify the mechanics of the physical restoration itself, the plan must also have a clear definition of injury, site selection, monitoring protocols and success. As mentioned earlier, those guidelines have been established (Fonseca, 1989, 1992, 1994), but have not yet been quantitatively coupled with the issue of interim loss to determine replacement ratios.

Recently, NOAA developed and implemented HEA using basic biological data to quantify interim lost resource services (NOAA, Damage Assessment and Restoration Program, 1997*a*). While sharing many of the same principles as other methods incorporating interim losses into replacement ratio calculations for wetlands (Unsworth & Bishop, 1994; King *et al.*, 1993), HEA focuses on the selection of a specific resource-based metric(s) as a proxy for the affected services (e.g. seagrass short-shoot density in the example discussed below), rather than basing its calculations on a broad aggregation of injured resources. Determination of this metric was one of the conclusions from the expert panel as discussed in Box 7.5 (biomass, as opposed to shoot density, has not yet been adopted because of a lack of empirical data on the recovery rate of belowground biomass, whereas recovery rate of shoots is a robust data set; this choice is an extremely generous concession to the responsible parties). This approach has the advantage of making HEA applicable not only to a wide range of different habitats, but to injuries to individual species as well (see Chapman *et al.* [1998] for a discussion of HEA applied to the calculation of compensation for historic salmon losses). Additionally, the selection of a resource-based metric allows for differences in the quality of services provided by the injured and replacement resources to be captured and incorporated into the replacement ratio (NOAA, Damage Assessment and Restoration Program, 1997*b*). Without specification of a quantifiable resource metric, analysis of the recovery of the resource following injury and/or the success of the restoration project may be difficult to evaluate precisely. For example, in the wetlands context, alternative metric specifications may lead to significantly different maturity horizons (Broome *et al.*, 1986) as well as the level of functional equivalence ultimately achieved by the restoration project (Zedler & Langis, 1991).

Box 7.1 Application of the habitat equivalency analysis (HEA)

An example of applying HEA to habitat restoration occurred in a recent federal court case to provide compensation for the loss of 1.63 acres of seagrasses (turtlegrass, *Thalassia testudinum*) within the Florida Keys National Marine Sanctuary (*United States of America* v. *Melvin A. Fisher et al.*, 1997). Extremely energetic hydrodynamic conditions at the injury site, together with intense grazing of the seagrass by nocturnal herbivores prevented successful establishment of seagrass plantings. Therefore, off-site restoration was chosen in the form of in-kind (same species) repair of *T. testudinum* beds previously damaged by boat propeller scars (prop scars). This approach focused initially on planting a native pioneering seagrass species, *Halodule wrightii*, to facilitate the eventual recovery of the slower-growing *T. testudinum*. This sequence, termed 'compressed succession' (M. Moffler, personal communication) promotes more suitable conditions for the slower-growing *T. testudinum* to encroach naturally upon the prop scar while temporarily stabilizing the site and preventing additional erosion with a more rapidly growing species (see Box 7.3). Project success was to be quantified by four parameters: (1) at planting, a minimum average of one horizontal *H. wrightii* rhizome apical per planting unit must be installed; (2) 75% survival of planting units at the end of year 1; (3) seagrass shoot density (as compared to nearby natural beds); and (4) achieve the target acreage of bottom coverage within a three-year monitoring period. Additionally, if monitoring indicated that performance standards were not being met or were not projected to be met, remedial plantings of those affected areas were designed into the plan. However, all remedial plantings reset the monitoring clock for that portion of the project. The ultimate success criterion was unassisted persistence of target bottom coverage by the seagrass plantings for three years, using photo-documentation to provide a common basis of assessment, perception, and historical reference.

Key factors in the National Oceanic and Atmospheric Administration (NOAA)'s development of restoration plans have been issues of pre-project planning, particularly regarding site suitability. Here, sites were reviewed for suitability using the following criteria: (1) they were adjacent to natural seagrass beds at similar depths; (2) they were anthropogenically disturbed; (3) they existed in areas that were not subject to chronic storm disruption; (4) they were not undergoing rapid and extensive natural recolonisation by seagrasses; (5) seagrass restoration had been successful at similar sites; (6) there was sufficient area to conduct the project; and (7) similar quality habitat would be restored as was lost. The restoration of seagrass prop scars created by vessel impacts represented NOAA's preferred approach to seagrass restoration off-site. In order to select a planting site that could accommodate the project's size, the amount of restored area was computed using the HEA.

Description of the compensatory restoration scaling approach

Accurate determination of the appropriate target scale of compensatory restoration[1] projects is necessary to ensure that the public and the environment are adequately compensated for the interim service losses. For injuries to seagrass resources, NOAA has employed HEA as the primary methodology for scaling compensatory restoration projects. The principal concept underlying HEA is that the public and the environment can be made whole for injuries to natural resources through the implementation of restoration projects that provide resources and services of the same type, quality and comparable value. HEA has been applied in cases centered on seagrass injuries because those incidents typically meet the three criteria defined by NOAA: (1) the primary category of lost on-site services pertains to the biological function of an area (as opposed to direct human uses, such as recreational services); (2) feasible restoration projects are available that provide services of the same type and quality and are comparable in value to those lost; and (3) sufficient data on the required HEA input parameters exist or are cost-effective to collect. If these criteria are not met for a particular injury, other valid, reliable approaches and methodologies are available for scaling the chosen compensatory restoration projects (NOAA, 1997*b*). These criteria for the use of HEA were upheld by the US District Court (*United States of America* v. *Melvin A. Fisher et. al.* 1997 92-10027-CIV-DAVIS). Of equal importance to the Mel Fisher decision was the decision by the US District Court in *United States of America* v. *Great Lakes Dredge & Dock Co.* 1999 97-2510-CIV-DAVIS to uphold the use of the HEA as a proper method by which to scale compensatory restoration.

At its most basic level, HEA determines the appropriate scale of a compensatory restoration project by adjusting the project scale such that the present value of the compensatory project is equal to the present value of interim losses due to the injury of that action (e.g., freshwater diversion projects intended to create wetland acreage).[2] This 'balancing' of gains and losses is accomplished through a four-step process (NOAA, 1997*a*). First (step 1), the extent, severity, and duration of the injury (from the time of the injury until the resource reaches its point of maximum recovery), and functional form of the recovery curve must be determined, in order to calculate the total interim resource service losses. Next (step 2), the resource services provided by the compensatory project over the full life of the project must be estimated to quantify the benefits attributable to the restoration. This step is analogous to the previous one and requires estimation of both the time required for the compensatory restoration project to reach its maximum level of service provision and the functional form of the maturity curve. After these resource service losses and gains have been quantified, the scale of the compensatory project is adjusted until the projected future resource service gains are equal to the interim losses associated with the injury (step 3). This process is depicted graphically in Fig. 7.1, where the scale of the compensatory restoration project is adjusted until the area under the maturity curve (the total resource service gains, represented by area B) is equal to the interim lost resource services (represented by area A). Because these services are occurring at different points in time, they must be translated into comparable present value terms through the use of a discount rate.

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[1]Compensatory restoration refers to any action taken to compensate for interim losses of natural resources and services that occur from the point of the injury until recovery of those resources/services to baseline. Conversely, primary restoration refers to actions that return the injured natural resources and services to baseline.

[2]In some instances, it may be beneficial to all parties involved to implement a project where the total discounted gains from the compensatory project exceed the total discounted losses. This situation occurs when the scale of the preferred project can only be adjusted according to a binary or stepwise function rather than a continuous function, or when the resulting amount of natural resources/services generated by a restoration action cannot be tightly controlled following implementation.

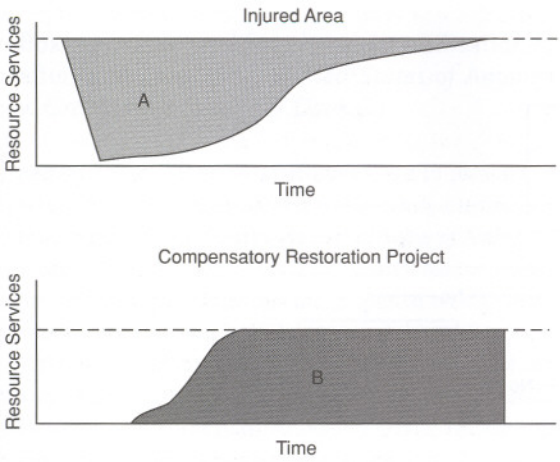

Fig. 7.1. Graphical depiction of how habitat equivalency analysis (HEA) sets the compensatory restoration to equal interim loss of resource services. This is achieved by setting the total services (hectare–years) lost until complete recovery back to pre-injury conditions (area A) is equalled by the services rendered under the compensatory project (area B).

Discounting is a standard economic procedure that adjusts for the public's preferences for having resources available in the present period relative to a specified time in the future. Because of discounting, plantings that occur longer after an impact are worth less in present-value terms than plantings conducted shortly after an impact, and therefore more planting must be done as time elapses. Finally (step 4), appropriate performance standards associated with the compensatory restoration must be developed to ensure that the project provides the anticipated level of services. Well-defined and measurable standards are essential to the success of the project regardless of whether the restoration will be implemented by the parties responsible for the original resource injury or by the management agency (trustees) using monetary damages which are recovered.

In Box 7.5, we present the outcome of a national workshop that set the stage for NOAA to provide reasonable and fair assessments of injuries to seagrasses and the effort needed to recover the lost resources which must be assumed by the responsible party.

The importance of site selection

Clearly one of the largest problems with seagrass transplanting is finding an appropriate place to conduct the restoration and install the plantings. It is not advised to plant seagrasses in areas with no history of seagrass growth, or where the aforementioned disturbances have not ceased. Planting should not be done under those circumstances because of the low probability of success. Planting may be done in open, unvegetated areas among patches of seagrass, but only for the goal of experimental manipulations and/or the evaluation of planting techniques (keeping in mind that these among-patch locations are not a strong test of the efficacy of a technique as they are embedded within viable seagrass territory). Seagrass patches migrate, alternately colonising currently unvegetated sea floor and dying out where seagrass is located presently (Marba *et al.*, 1994; Marba & Duarte, 1995; Fonseca *et al.*, 1998*a*, 2000*b*). Thus, the spaces between the patches today may be naturally colonised by seagrasses in the future.

Campbell *et al.* (2000) provide a decision strategy for assessing the selection planting sites that include measures of light, epiphytisation, nutrient loading, water motion, depth, proximity of donor site and alternative actions (Fig. 7.2). Similarly, Fonseca *et al.* (2000*a*) and Calumpong *et al.* (in press) give the following criteria for the selection of a restoration site away from the original injury site:

- It is at depths similar to nearby seagrass beds
- It was anthropogenically disturbed
- It exists in areas that are not subject to chronic storm damage
- It is not undergoing rapid and extensive natural recolonisation by seagrasses
- Seagrass restoration has been successful at similar sites
- There is sufficient area to conduct the project
- Similar quality habitat would be restored as was lost.

These selection criteria have been used successfully in the US Federal Court as the basis for seagrass restoration projects (*United States of America* v. *Melvin A. Fisher et al.* 1997). By considering these

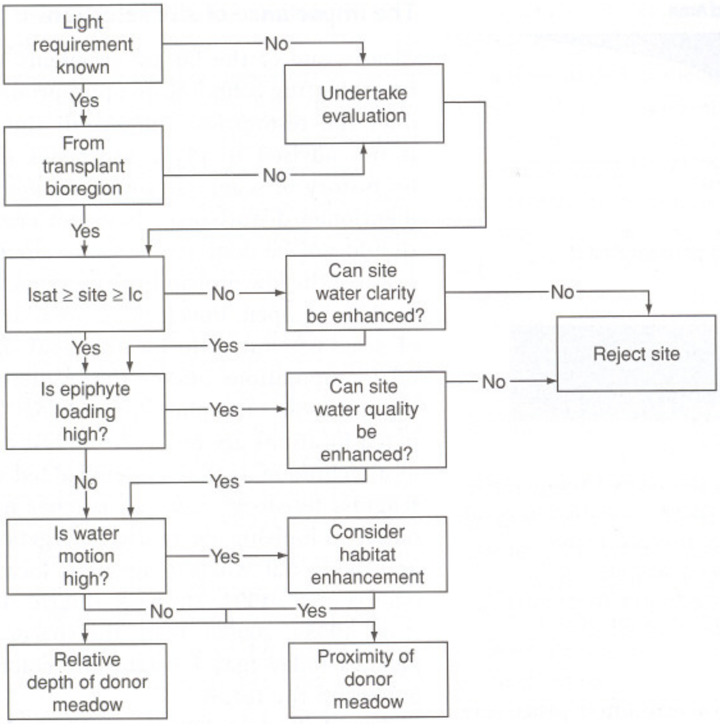

Fig. 7.2. Decision flow diagram regarding site selection for restoration. From Campbell *et al.* (2000). Isat = saturation irradiance, Ic = compensation irradiance.

criteria, it is apparent that transplantation should probably not be undertaken for the purposes of enhancing recovery from natural disturbance events as these events have both an ecological and evolutionary function in determining the survival and fitness of the seagrass ecosystem. When possible, rehabilitation of the primary injury site should be performed to restore or accelerate the recovery of baseline service flows, with compensatory restoration used to compensate the public for interim service losses that accrue while the site reaches its pre-injury levels of service provision.

Critical factors influencing transplant success

Numerous factors have been determined to affect transplanting success. Some are of the crop-risk type, are extrinsic and cannot be controlled. Others involve issues of protocol. In a survey of North American seagrass planting projects, Fonseca *et al.* (1998*a*) listed the following as factors that had the potential of being controlled by those conducting transplants:

- Similarity of environmental conditions of donor and recipient beds.
- Choice of species: preferably same as that lost, but pioneering species may be substituted to initiate a project.
- Presence of grazers or sediment burrowers: these bioturbating organisms may need to be excluded or plantings may have to be conducted in large patches to dissuade them from their activities.
- Source of planting material: similar depths and environmental conditions and from over as broad a geographic area as possible to ensure genetic diversity.
- Time of year: seagrass should be planted at a time to ensure the longest period before seasonal stressors.
- Cost: many variations of cost have been given but standardised costs are elusive; based on recent cases in the US federal court, a contracted project that

Box 7.2 The need for physical stabilisation after boat grounding: *United States of America* v. *Great Lakes Dredge & Dock Co.*

Another example of the difficulties faced in primary restoration of seagrass habitat was recently illustrated in another Federal Court case (*United States of America* v. *Great Lakes Dredge & Dock Co. and Coastal Marine Towing Inc.* 1999) where a large tugboat grounded on a shallow seagrass–*Porites* coral bank in 1993 and destroyed 7200 m^2 of habitat. The grounding site was located in an exposed, high-energy environment where seagrass transplantation was deemed inappropriate. The expert case team assembled by NOAA and the US Department of Justice recommended that the primary restoration plan should include filling and regrading the trench made by the tugboat to physically stabilise the site. It was assumed that once the site was physically stable the seagrasses would recolonise naturally, but slowly. Interim losses of seagrass would be compensated off-site in a plan similar to the plan for the *United States of America* v. *Melvin Fisher et al.* described previously. Attempts to negotiate a settlement with the responsible parties proceeded for five years without a resolution. In September 1998 the site was impacted by Hurricane Georges which severely damaged the partially recovered portions of the injury and effectively set back the recovery clock (Whitfield *et al.*, in press). Seagrass beds adjacent to the injured area were unaffected. The impact of the hurricane confirmed initial concerns that the grounding site was physically unstable and vulnerable to further injury. Clearly, there are cases in high-energy environments where physical restoration is needed to stabilise injured sites to promote the recovery of seagrasses. In the primary restoration plan for this case, the amount of sediments excavated by the grounding called for substantial *in situ* engineering to recreate the bank structure. This is not surprising, since it took nature between 500 and 1000 years to form the bank that was destroyed. Recent studies have shown that large vessel groundings are becoming more common on seagrass banks in south Florida and elsewhere in the Caribbean (Whitfield *et al.*, in press). Whitfield *et al.* (in press) have documented the instability of these injuries and recommend that physical regrading is necessary to prevent further damage during severe storms.

includes site surveys, planting, monitoring and reporting will cost (in 1996 US dollars) ~US$630 000 per hectare.

It is essential to study the substrate-energy (exposure) regime, and optical water quality (clarity or light availability) of the area that will be transplanted so that suitable source materials can be identified. Areas exposed at low tides should be carefully mapped so as to place plantings with minimal exposure to air, unless the plants are regularly occurring in the intertidal zone (e.g. in the Pacific Northwest of the United States). Planting in high wave energy or tidal current areas will require planting in larger groups to avoid disruption (Fonseca *et al.*, 1998*a*, *b*). Planting in larger groups also appears to be an effective method of deterring physical disruption of the planting by marine organisms. However, as suggested by Addy (1947), matching water depths, temperature, salinity, water clarity and plant size remain some of the best general guidelines for matching donor and recipient beds.

The characteristics of the species, such as fast growing vs. slow growing; pioneering vs. climax, annual vs. perennial growth, etc. must be considered before conducting transplantation work. For example, *Halodule* spp. and *Halophila* spp. are fast-growing pioneering species while *Thalassia* spp. and *Enhalus* spp. are slow-growing climax species. *Halophila* spp. rapidly colonise disturbed areas like those with moving sand bars and are under-canopy species, requiring low light. Although a climax species may have been disturbed, it is often advisable to first install a faster-growing species to stabilise the environment.

Another important factor in the selection of seagrass for transplanting, besides their intrinsic recovery rate, is their growth habit (Short & Short, 2000). Transplanting can be rendered almost wholly ineffective if meristematic regions of these plants are

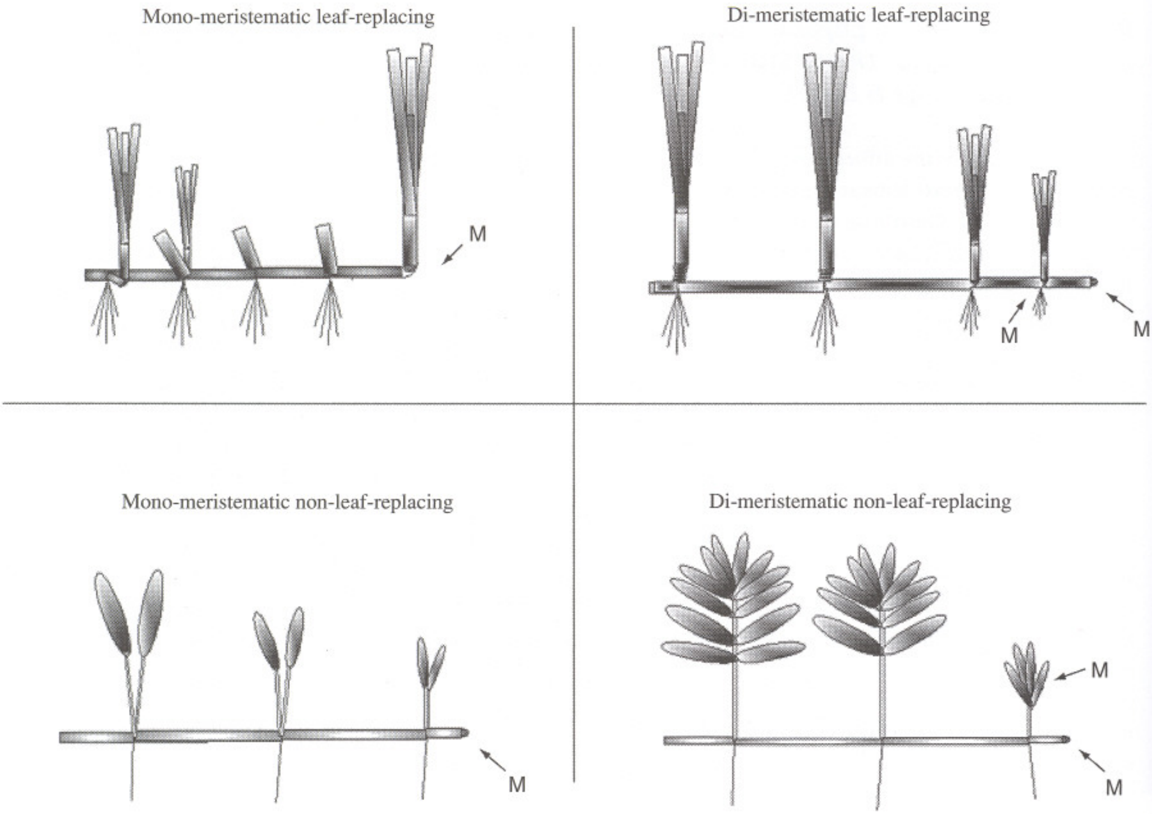

Fig. 7.3. The four basic growth forms of seagrass. In the case of the mono-meristematic, leaf-replacing form, each terminal shoot on the runner is a viable planting unit – comparatively low modular integration is present meaning that most often each shoot has high potential for contributing to spatial colonisation. The other three forms require at least three to four short shoots be maintained on the runner for a complete planting unit, as well as an intact rhizome apical meristem. From Short & Short (2000).

damaged or not incorporated in sufficient quantity in a planting unit to initiate recolonisation. Short & Short (2000) summarise the morphotypes of seagrass (Fig. 7.3).

Seagrass grazers can have disastrous effects on plantings. Seagrass grazers include sea-urchins, gastropods and herbivorous fishes. Some migratory waterfowl such as geese and ducks have been observed to decimate seagrass plantings (personal observation). Significant grazing of natural *Syringodium filiforme* beds points out the general susceptibility of seagrasses to grazing (Rose *et al.*, 1999). Fonseca *et al.* (1994 and references therein) found significant disturbance by rays in Tampa Bay, Florida, indicating that it would be necessary to use exclosure cages to ensure the survival of transplanted seagrasses in some areas. Recently, we have seen that planting in clumps of at least 20–50 cm on a side deters many animals from disturbing the plantings (authors' unpublished data).

Minimisation of disturbance to the source bed is paramount in seagrass transplanting so as not to exacerbate injury to local populations. With present techniques focusing on the use of wild, vegetative stocks, this may be achieved by conducting the transplantation in phases, or dispersing the collection effort, thus allowing the source bed to recover. Harvesting of donor stock should also be done from

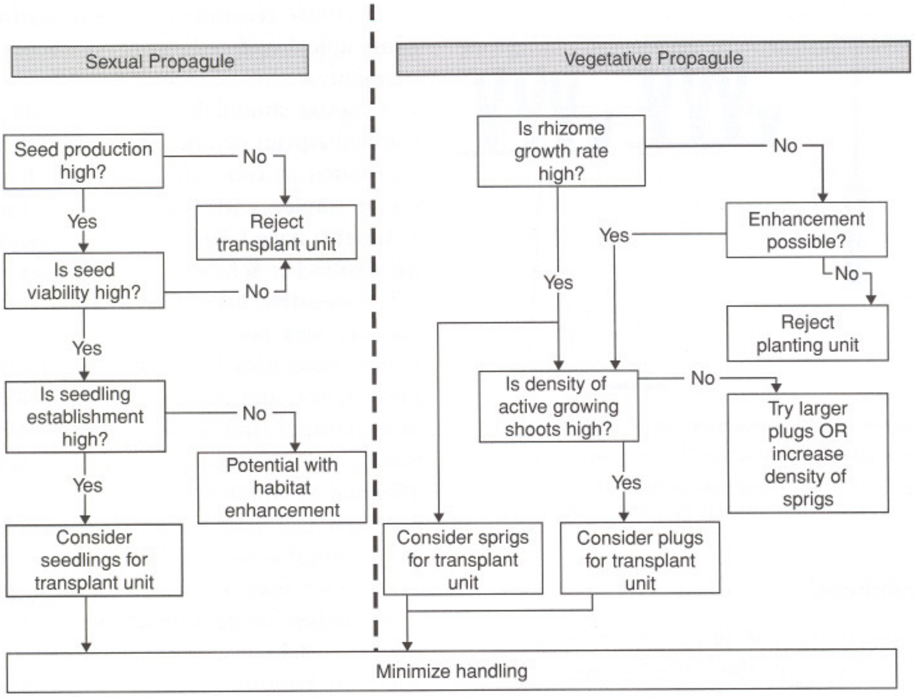

Fig. 7.4. Decision tree for choosing seedlings or whole, mature plants for transplanting. We caution that the technology for seed establishment is not as developed as for the use of sprigs or cores. From Campbell *et al.* (2000).

beds over as broad a geographical area as possible. This may help avoid loss of genetic diversity in the planted bed (*sensu* Williams & Orth, 1998) and may actually incorporate the full local range of genetic diversity into the planting.

Fortunately, many pioneering species can be harvested with minimal disturbance to the beds (Fonseca *et al.*, 1994). However, for climax species, harvesting of donor beds may cause long-lasting damage and harvesting from these beds should only occur when the beds are under some anthropogenic source of physiological stress that does not seem likely to abate or if they are in imminent danger of physical removal (e.g. dredging).

The size of the source or donor bed should first be assessed to determine if recovery will proceed after removal of the sods, cores or sprigs. This is especially true when transplanting vegetative stock, as a large amount of material is needed. Spacing harvesting at ~0.25 m for small cores or sods (<0.15 × 0.15 m) is often sufficient to avoid long-lasting damage. Moreover, Fonseca *et al.* (1998*a*) suggested that *Ruppia, Halophila, Halodule* and *Zostera* spp. can recover in small patches (<0.25 m^2) within a year with shoot density returning to normal. Furthermore, Fonseca *et al.* (1998*a*) cautioned that patches >~30 m^2 in high-current areas may never recover. Campbell *et al.* (2000) also provide a decision tree for selection of planting stock for both sexual and asexual propagules that focuses on intrinsic propagation rates (Fig. 7.4).

Choosing the time-frame for planting is an obvious concern, and as with all crops, the appropriate time for seagrass varies with geographical region. In general, the best strategy is to plant at a time just after the period of highest seasonal stress, when natural populations are experiencing recovery. For example, eelgrass (*Zostera marina*) should be planted in the autumn in North Carolina, and other mid-Atlantic regions in the United States, because summer is the period of maximum physiological stress at that location (Moore *et al.*, 1997).

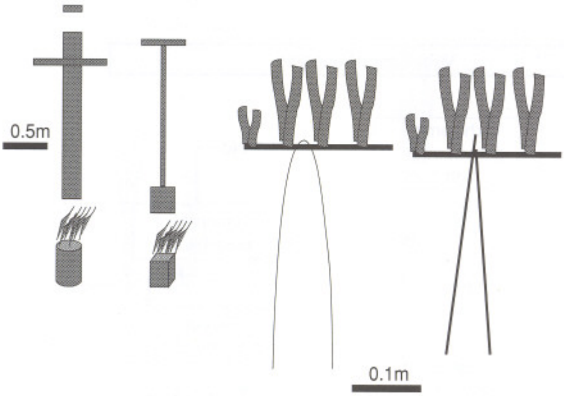

Fig. 7.5. Diagrammatic representation of the two most widely used seagrass planting methods; those with sediment such as cores and plugs and those without sediment, usually anchored with metal or wood staples.

Planting methods

Fonseca *et al.* (1998*a*) list 14 categories of planting methods for seagrass in the United States. For these methods, source material can be vegetative stock or seeds. Transplantation using vegetative stock typically requires available wild stock as a source and is labour-intensive and invariably expensive. However, it often gives faster, more reliable coverage than seed methods (but see review by Orth *et al.*, 2000). Most projects today are carried out using either small sods or sprigging of sediment-free units (Fig. 7.5).

Sediment-free methods

For most sediment-free methods, plants are dug up using shovels, the sediment is shaken from the roots and rhizomes and the plants are placed in flowing seawater tanks or floating pens. For vegetative stocks, Fonseca *et al.* (1998*a*) recommend a minimum of one apical shoot per planting unit. The number of short shoots on a long shoot should be maximised whenever possible, so as to derive benefits from the clonal nature of the plant. Also, the plants should be collected and planted on the same day, kept in water with the same ambient temperature and salinity, and kept as moist as possible when out of the water. When using vegetative stocks of *Thalassia testudinum*, Tomasko *et al.* (1991) recommend a minimum of one rhizome apical and at least three shoots per rhizome segment.

Seagrass should be planted either directly into the bed (sprig) or anchored using a variety of devices such as rods, rings, nails or Rebar. U-shaped metal staples with attached bare root sprigs (no sediment) have been widely used as planting units (Derrenbacker & Lewis, 1982; Fonseca *et al.*, 1982) or, when negative buoyancy is not required, bamboo skewers may be substituted (Davis & Short, 1997). Plants have also been woven into biodegradable mesh fabric and attached to the sediment surface as a planting unit (Fonseca *et al.*, 1998*a*). Rocky intertidal species, such as *Phyllospadix* spp., have been attached to boulders.

When using anchoring devices, one must consider using biodegradable or natural materials such as boulder over metal or plastics. As mentioned above, when using staples, one can choose metal (US$0.01 each) or can modify 'shish kebab' bamboo sticks by bending them into a V (Davis & Short, 1997) which when purchased in bulk could cost only US$0.006 a piece. In tropical areas where bamboo is plentiful, this could be a more economical medium to use. Bamboo is also biodegradable. Using either kind of staple, planting units are made by grouping plants and attaching the root–rhizome portion under the bridge of a staple and securing the plants with a paper-coated metal twist-tie. This can either be prepared beforehand or the planting unit can be pinned directly to the substrate during planting.

When using nails, boulders or Popsicle sticks (Merkel, 1988), the technique is more or less similar and the planting unit is tied to the anchoring instrument. Frames, such as Short's TERF device (F. T. Short, in Fonseca *et al.*, 1998*a*), have great promise for rapid and non-diver-assisted planting at depth. A cage deployment system that has shoots attached to the bottom is lowered onto the sea floor and retrieved after the shoots have rooted and their paper ties have decomposed. This eliminates the need for divers in deeper water, can be used in chemically polluted areas, and provides initial protection of the plantings from biological disturbance.

Seagrass with sediment methods

The sod or turf method consists of planting a shovelfull of seagrass with sediment and rhizomes intact. This is the easiest method, and is most applicable for hard, compact substrates and deep-rooted and large species such as *Enhalus acoroides*. The only equipment needed are shovels and large basins for the sods. However, if the donor site is far away, transporting the sods may present a problem as the weight of the material is a physical burden. Some species, such as *E. acoroides*, *Posidonia* spp. and *Thalassia* spp. may have very deep root–rhizome systems requiring removal of a tremendous amount of sediment to harvest the belowground plant structures all intact (Fig. 7.6). To our knowledge, this has only been accomplished in Western Australia (by E. Paling, of Murdoch University; see review in Fonseca *et al.*, 1998*b*). Furthermore, harvesting an entire sod may constitute one of the most severe perturbations in a seagrass meadow, inhibiting recovery in the donor bed.

The plug method utilises tubes as coring devices to extract the plants with the sediment and rhizomes intact. The plugs are planted directly into the seagrass bed after creation of a hole to receive the contents of the tube. The core tubes are usually made of 4–6-cm diameter PVC plastic pipe with caps for both ends to initially create a vacuum and keep sediments from washing out the bottom. The tube is inserted into the sediment, capped (which creates a vacuum), pulled from the sediment and capped at the other end to avoid losing the plug. This can only be done with soft but cohesive sediments and generally only for small species to avoid excessive leaf shearing (unless extreme care is taken to avoid the shearing, which adds measurably to the cost of the process). When the donor bed is far away from the planting site, many tubes are needed which also adds to the cost.

Sod pluggers extract a plug out of the donor bed which is then extruded into a peat pot; the method was first used by Robilliard & Porter (1976)

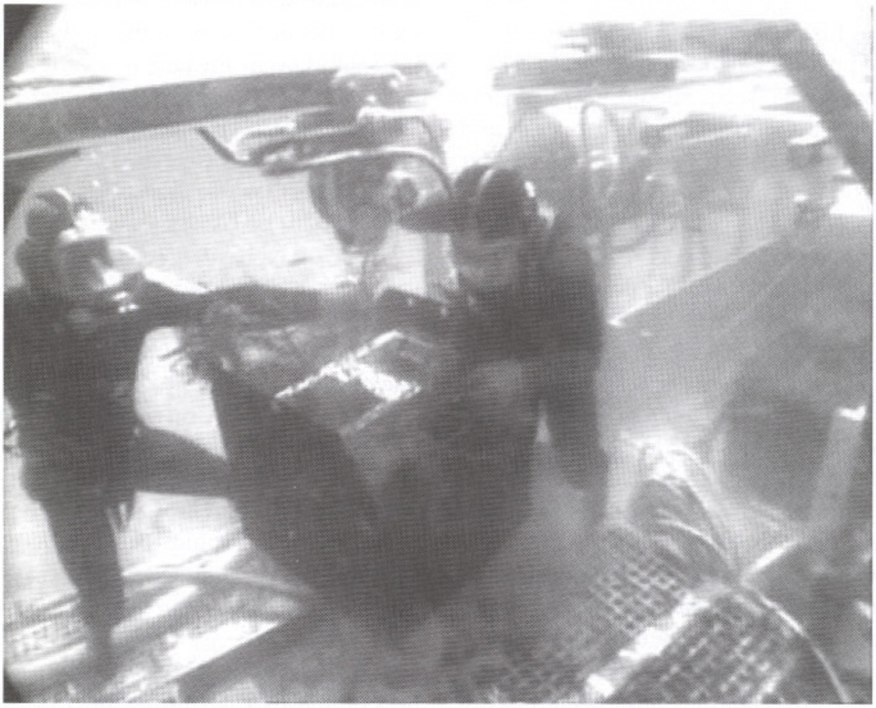

Fig. 7.6. Harvest of *Posidonia* sod near Perth, Western Australia. Photo courtesy E. Paling.

and modified by Fonseca *et al.* (1994). Because these pots are typically only a few cm across, they may be inserted into the bottom by liquefying the sediment with a hand tool. After the peat pot is planted, its side walls must be ripped off or torn down and the pot pushed into the sediment to allow the rhizomes to spread out.

Sowing of seed

Seed planting holds promise for large-scale restoration but is currently more applicable only in low-energy areas where the seeds can settle and germinate and where there are few seed predators. This method was first introduced by Thorhaug (1974) with *Thalassia testudinum*. A seedling grow-out method for *T. testudinum* has been registered by Lewis (1987). The availability of seeds must also be considered. Large areas in the Chesapeake Bay have been established by sowing seeds from a small boat (R. J. Orth, personal communication). Work continues in this highly promising area (Orth *et al.*, 2000). Experiments using seeds pelletised to increase their density to facilitate sinking and seeds embedded in biodegradable mesh are presently being carried out by Granger and his colleagues (Granger *et al.*, 2000). Experiments on planting depth also indicate that at least for *Zostera marina*, seeds should be within the top 2 cm of the sediment for best germination and that sowing densities should be 400–1000 seeds per square metre (Granger *et al.*, 2000).

Laboratory cultured stocks

This approach uses plants reared and grown in the laboratory from plant fragments. It may become especially applicable for large-scale plantings where a large amount of planting units is needed. This technique also holds promise for reducing or eliminating donor bed damage and this has been shown to be minimal for pioneering species, such as *Halodule wrightii* and *Syringodium filiforme* (Fonseca *et al.*, 1994). This approach also has the potential to maintain donor stocks for unscheduled plantings and could theoretically supply genetically variable and disease-resistant plants.

Several aspects of this approach remain controversial. So far, three species have been successfully propagated in the laboratory, *Ruppia maritima, Halophila decipiens* and *H. engelmannii* (M. Durako, personal communication). *Ruppia maritima* has been successfully transplanted from laboratory culture stock (Bird *et al.*, 1994), but all these species are naturally fast growing (i.e. pioneering) and it is unclear whether laboratory culture is a cost-effective means of restoring naturally prolific species. Moreover, questions regarding the ability to maintain genetic structure of the population have not been solved. Given the growing emphasis on mechanised plantings using wild stock, laboratory culture will probably only be cost-effective when techniques are developed for slow-growing species, hence avoiding long-term donor bed impacts.

Monitoring the restoration

Monitoring of the restoration project is necessary to provide data required to evaluate the viability of the project based on the performance standards (defined below). This permits timely identification of problems or conditions that may require corrective action to ensure the success of the project.

Monitoring schedule and activities

Field collection of data for performance monitoring should occur for four years after planting. Original plantings should be monitored for three years and potential remedial plantings in year 2 should be monitored for three years for a total monitoring period of four years. Under this schedule the monitoring would be conducted as follows:

year 1 – day 60, 180, 365
year 2 – day 180, 365
year 3 – day 180, 365
year 4 – day 180, 365

The precise dates are weather-dependent. In carbonate sediments, each surviving planting unit should receive an additional spike of constant-release phosphorous fertiliser (0–39–0, nitrogen–phosphorus–potassium) at day 60 of year 1. Alternatively, bird roosting stakes could be installed about every 5–10 m along scars (see Box 7.3).

Data collection

Monitoring should focus on documenting the numbers of apicals at planting time, planting unit survival, shoot density and areal coverage under the following schedule and definitions. This monitoring

Box 7.3 Transplanting strategies

RESTORING SLOW-GROWING SPECIES

Durako *et al.* (1992) developed a strategy for restoring slow-growing species such as *Thalassia testudinum* by mimicking natural succession. They initially plant another, faster-spreading congener, such as *Halodule wrightii* to achieve 'compressed succession' (see Box 7.1). This temporary substitution of a faster-growing species for a slow-growing one promotes a more suitable condition for the slow-growing one to establish itself while the faster one stabilises the sediment and provides a functional seagrass habitat.

Based on a series of ecological field experiments (Fourqurean *et al.*, 1995) and transplanting studies (Kenworthy *et al.*, 2000), we have determined that 'compressed succession' of *H. wrightii* can be enhanced by fertilising transplants with bird roosting stakes. The procedure works by placing bird roosting stakes over the transplants. Seabirds roosting on the stages defecate nutrients into the water, stimulating rapid growth of *H. wrightii* transplants. In cases where there is a mixed meadow of *T. testudinum* and *H. wrightii* recovery of an injured area can be accelerated by placement of stakes alone.

STEPS IN A GENERALIZED PROTOCOL

- Study the site to be restored and determine at least the following parameters; (a) seagrass bed history (species composition, cause of loss); (b) exposure to air and waves and currents; (c) substrate type – avoid clays and high organic sediments; (d) rate of siltation – plants often cannot withstand much more than 25% burial and vertical growth is possible only for some species; and (e) presence of animal disturbance.
- Determine time-frame and budget by evaluating the typical staffing requirements. Merkel (1992) estimated a minimum of seven persons were required for intertidal, bare-root (e.g., staple technique) and nine persons for subtidal bare-root planting. The use of scuba incurs higher costs that need to be factored in. In the Philippines, Calumpong *et al.* (1992) accomplished planting of 32 0.125 m^2 sods in the inter- and subtidal with nine persons in one day. Fonseca *et al.* (1994) provide a comparison among several methods using timed trials. They found that collection + fabrication + planting costs ranged from ~1.2 to 3.5 minutes per planting unit with peat pot plugs being the most rapid method and cores being the slowest method.
  It is imperative that one recognises what is not timed in that report, such as mobilisation and demobilisation, both daily and on a project-scale basis, travel, reporting and monitoring. The timed trials also did not accurately measure the effect of boredom on the speed of the process.
- Locate a donor bed that matches the conditions at the planting site. For vegetative methods, this should be near enough so the shoots can be planted the same day. Overnight storage of material, particularly bare-root material, should be placed in moving seawater at ambient temperatures and salinities. To our knowledge, there has not been a sufficiently controlled experiment to determine the storage capability for seagrasses.
- Be prepared to manage the workforce with regard to the tedium of tasks. To avoid boredom, varying tasks among individuals can be a useful strategy.
- Carefully delineate plots to facilitate monitoring.
- Consider all the aforementioned potential costs. These include site delineation, reports, mobilisation and demobilisation, insurance, overhead, benefits, mapping, planting operations, monitoring, remedial planting and a 10% profit margin for contractors.
- Conduct thorough monitoring (see below) and be prepared to conduct remedial plantings.

protocol applies to original plantings for three years (years 1–3) and to remedial plantings for three years (years 2–4).

1. Apical counts. Prior to planting, one planting unit out of every 100 collected should be examined for the number of rhizome apicals.
2. Survival. Each site should be examined for survival of all planting units during each survey in year 1 (days 60, 180 and 365) or until coalescence. Survival of each species should be expressed as a percentage of the original number, but the actual whole number should also be reported.
3. Shoot density. A separate (from survival) random

selection of three planting units per 100 planted should be assessed for number of shoots per planting unit at each survey time until coalescence begins. After some planting units begin to coalesce, three randomly selected locations per 100 m^2 (100 planting units) should be surveyed for shoot density over a 1 m^2 area at 0.0625 m^2 (25 cm×25 cm) resolution. Shoot density should be monitored for three years.

4. Areal coverage. The randomly selected planting units (may be the same as shoot density selection) should be surveyed for coverage at each survey time starting at day 180 of year 1. Measurements should be taken at a 0.0025 m^2 (5 cm × 5 cm) resolution prior to coalescence and over a 1 m^2 area at 0.0625 m^2 (25 cm × 25 cm) resolution after coalescence for each seagrass species present at each survey time. Areal coverage should also be monitored for three years.
5. Video tape transects. Five 100-m transects along randomly selected portions of the planted area should be video tape recorded to establish permanent visual documentation of the progression of areal coverage of seagrass through time. A tape measure should be laid along the central (long) axis of the scar and should be included in the video tape to allow physical reference of locations within the scar. Video recordings should be taken at each survey time during the monitoring period of three years. Observation-based assessment of success may be substituted if quadrats are used in accordance with a Braun–Blanquet survey method (Fonseca *et al.*, 1998*a*) or if the data are obtained from the video tape (making the observational data base available for cross-checking). The same number of sample points must be obtained with the same spatial extent (i.e. survey each scar). Similarly, Braun–Blanquet observations of cover at every metre along each scar may also be obtained from the video tape to obtain estimates of planting performance.

Reporting requirements

Monitoring reports should include copies of raw data gathered in each survey, an analysis of the data, and a discussion of the analysis. Originals of all video tapes recorded since the previous report should be provided with each new report. Originals of all video tapes and other photography should be turned over to the permitting agency following project completion by the party conducting the monitoring.

Remedial plantings and/or project modifications

If data from a monitoring report establishes that the performance standards are not being met or are projected not to be met, remedial plantings of those

> **Box 7.4** Performance standards
>
> Although it is the overall objective to restore the species that was injured, performance criteria may also be based on the success of planting of pioneering seagrass species, as found when *Halodule* is installed to expedite the recovery of *Thalassia* (Fonseca *et al.*, 1998*a*).
>
> **APICALS**
>
> A minimum average of one horizontal rhizome apical per unit should be maintained in all original planting and remedial planting.
>
> **SURVIVAL**
>
> The survival rate shall be considered successful if a minimum of 75% of the planting units have established themselves by the end of year 1. If it is determined that less than 75% survival has occurred by the end of year 1, then remedial planting should occur during the next available planting period to bring the percentage survival rate to the minimum standard by the next monitoring survey.
>
> **GROWTH**
>
> The third success criterion should be the measured growth rate of bottom coverage. The growth rate should be considered successful if, starting after one year, the planting is projected to achieve the total desired acreage of bottom coverage, with 95% statistical confidence, within the three-year monitoring period for original plantings. If this criterion is not met, then remedial planting should occur during the next available planting period.

affected seagrass species should occur. If there is a recurring problem with survival of plantings or replantings in a particular area, remedial planting should occur in another suitable area in as close proximity as possible, subject to the approval of permitting agencies.

Based on past experience in seagrass restoration efforts, it is assumed that 30% of the planted area should require remedial planting in year 2. All original plantings should be monitored for three years. Remedial plantings should also be monitored for three years.

Box 7.5 National workshop for defining the metrics of assessment for seagrass restoration projects

National Oceanic and Atmospheric Administration (NOAA) is designated as a trustee for natural resources under several United States laws,[1] and, in that capacity, is authorised to act on behalf of the public to seek compensation for injuries to its trust resources. Under each of these statutes, natural resource damage claims are composed of three basic components: (1) the cost of restoring the injured resource to baseline; (2) compensation for interim lost resource services[2] from the time of the injury until restoration occurs; and (3) the cost of performing the damage assessment. Habitat equivalency analysis (HEA) is one of the more frequently used methodologies available to natural resource trustees for calculating the appropriate scale of restoration projects necessary to compensate for interim resource service losses (Fonseca *et al.*, 2000*a*). The basic approach underlying HEA is to determine the amount of compensatory habitat to be restored, enhanced and/or created, such that the total services provided by the compensatory project over its functional lifespan are equal to the total services lost due to the resource injury.[3]

While HEA is conceptually and computationally straightforward, proper application of this approach requires a detailed understanding of the biological and ecological processes that affect the recovery and productivity of injured and restored habitats. In order to gain a better understanding of these processes, NOAA is undertaking a systematic, expert review of the ecological assumptions made within the HEA framework for a number of habitats for which HEA is most frequently applied. Seagrass habitats were selected as the first habitat to review for two primary reasons: (1) NOAA expects that due to the frequency of injuries to seagrasses (more than 70 000 hectares of injured seagrass in Florida alone), HEA will be commonly applied in cases involving injuries to seagrass habitats; and (2) the relatively small number of species of seagrasses present within areas under NOAA's trusteeship made this habitat a logical starting-point from which to develop and refine the review process for more diverse, complex habitats.

WORKSHOP PARTICIPANTS

Under the direction of NOAA's Southeast Fisheries Science Center, several academic experts in addition to NOAA staff were selected to discuss the underlying assumptions of HEA for seagrass environments. The workgroup was assembled to be geographically diverse, as well as to reflect a range of specialties within the field of seagrass biology/ecology. The workgroup participants were: Susan Bell, University of South Florida; Kenneth Moore, Virginia Institute of Marine Science; Mary Ruckelshaus, Florida State University; Frederick Short, University of New Hampshire; Charles Simenstad, University of Washington; Mark Fonseca,

[1]Notably, the Comprehensive Environmental Response, Compensation, Liability Act, the Clean Water Act, the National Marine Sanctuaries Act, and the Oil Pollution Act of 1990.

[2]Services here refer to the functions that one resource performs for another or for humans.

[3]For a detailed discussion of HEA, see National Oceanic and Atmospheric Administration, Damage Assessment and Restoration Program (March 1995, revised October 2000), Habitat Equivalency Analysis: An Overview, unpublished report.

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WORKSHOP CONCLUSIONS

Prior to the workshop, all participants were provided with background materials on the theory and application of HEA, as well as a series of null hypotheses developed by NOAA to capture the major issues relative to implementation of HEA for seagrass environments. Each of the null hypotheses discussed is presented below, accompanied by a summary of the conclusions reached by the workgroup.

Null hypothesis 1: Recovery of functional attributes can be forecast based on seagrass biomass and density alone[4]

The general conclusion of the workgroup was that seagrass biomass represents a more comprehensive metric of habitat function than seagrass shoot density. In general, biomass was cited as preferable to density measures because density measures do not capture the belowground component of seagrass systems, which may be important in determining the long-term persistence of recovering systems (particularly for *Thalassia testudinum*); and seagrass shoot density develops much more rapidly than function, and may overshoot baseline shoot density before achieving equilibrium.

Canopy volume (shoot density times the height of the seagrass canopy) was proposed as a preferable measure to biomass. While in most cases canopy volume would be expected to be highly correlated with total (above- and belowground) biomass, canopy volume has the added advantage of being easily measured in a non-destructive manner. Despite the apparent advantages of the canopy volume measure for capturing within-patch functions, the workgroup cited among-patch attributes of landscape structure, scale and setting that would also be expected to significantly influence functional performance and recovery rates. Cover and connectedness among vegetated patches were among the measures cited as important in capturing the among-patch aspects of seagrass habitats.

Null hypothesis 2: Forecasting seagrass bed recovery is independent of scale

The general conclusion of the workgroup was that seagrass bed recovery likely will be influenced by the spatial scale of the injury. One rationale provided for rejecting this hypothesis was that current data suggest seagrass beds modify their environment, and thus the size, severity and shape of an injury will affect the recovery process. Other participants suggested that as the scale of the injury increases, landscape features will be increasingly important in determining and restoring functional attributes of seagrass beds. Stated differently, participants expected differences in the recovery of a continuous seagrass bed versus a bed of the same total density or biomass, but distributed in discrete patches.

Null hypothesis 3: Overcompensation responses by seagrass in injured areas (e.g. generation of shoot densities higher than un-impacted controls) does not constitute enhanced ecological/biological functions

The general conclusion of the workgroup was that overcompensation responses by seagrass in injured areas do not constitute enhanced functions. Higher seagrass density does not necessarily indicate higher production. In addition, external controls, such as light availability, will serve as limiting factors on the function of a particular habitat. The workgroup also concluded that the presence of an observed overcompensation response may be an artifact of density-based measures of recovery, while biomass-based measures for the same area may not exhibit the same response.

The outcome of this workshop set the stage for NOAA to provide a reasonable and fair technical basis to assess seagrass injuries and support the recovery of the lost resource services for which the responsible party is liable.

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[4]Functional attributes refer to the range of ecological services provided by seagrass habitats including, but not limited to, primary production, faunal use, nutrient filtration, and sediment stabilization.

Table 7.1. *Top: General distribution (%) of costs by task (*United States of America *v*. Salvors Inc.*); bottom: summary of costs by specific actions (Fisher Natural Resource Damage Assessment claim)*

Task	Percentage of total costs
Map and ground-truth	5.5
Planting	18.5
Monitoring	58.7
Contractor	8.3
Government oversight	9.1

Type of cost	US$ (1996 values)
Damage assessment costs	
Federal assessment costs (up to 26 October 1996)	211 130
Interest on federal assessment costs at judgment	26 553
Subtotal	237 683
Restoration costs	
Primary restoration costs (vessel-generated holes in sea floor – restoration deemed not feasible with current technology)	0
Restoration site selection analysis	5 465
National Environmental Policy Act compliance/permitting costs	14 695
Preparation of map/ground-truthing sites	14 314
Collection, preparation and installation of planting units	64 846
NOAA restoration oversight/supervision costs	17 650
Subtotal	116 970
Monitoring costs	
Monitoring of compensatory prop scar areas	205 650
Contractor profit on restoration/monitoring work	29 028
Grand total for claim	589 331

Costs of restoration

From our experience, there is a general set of factors that drive up the cost of seagrass transplanting, particularly inappropriate site selection, inexperience, and disturbance events (requiring remedial planting). Consistent estimates of planting costs in dollars remain elusive, but recent restoration plans in the United States that have been litigated in the federal courts have shown the full cost of a restoration distributed among the various tasks (Table 7.1) at ~US$590 000 for a 1.55 acre area or ~US$940 000 per hectare (1996 dollars). Two important points here are that: (1) the actual costs of collecting and installing planting units is less than 20% of the actual cost of the entire project; (2) while monitoring costs at first glance may appear high relative to planting costs, it is important to note that monitoring represents a labour-intensive, multi-year effort to ensure that performance standards are met and necessary mid-course corrections are undertaken. The majority of planting costs on the other hand are incurred at a single point in time. This cost pattern is not unique to seagrass projects, but is commonly observed in natural-resource restoration projects across different types of habitats. We consider these data to be much more indicative of the real costs of executing a restoration project than previously presented (e.g. Fonseca *et al.*, 1982).

CONCLUDING REMARKS

In this chapter we have dealt with what we consider to be some of the critical issues that must be addressed in the implementation of effective restoration projects. These issues include: (1) choice of an appropriate metric, representative of the array of services provided by a resource, by which to measure success; (2) evaluation of lost interim resources; (3) appropriate selection criteria for off-site restoration projects; and (4) accurate project cost estimation. A fifth issue presented itself as we edited the paper – the role of disturbance. Disturbance is a fundamental ecological process and we noticed that it repeatedly worked its way into our discussions, signalling its obvious but subtle role in influencing the outcome of restoration projects. Finally we review methods, but we do not view these to be a weak link in the process, *per se*. The weakness in methods arises when workers do not study past efforts. Rather, failure of restoration arises in general from not considering the broader context of ecological injuries, particularly issue (3). When restoration plans are sent to us for consideration, the first aspect of the plan that we look at is the choice of a restoration site. Almost without fail for those with little restoration experience, a site is selected that is not damaged and does not need repair (e.g. planting in spaces among naturally patchy seagrass).

In the United States as elsewhere around the world, we have largely won the battle to recognise the value of seagrasses as a national resource. However, the acceptance by US federal courts of our metrics for assessing success, the concept of interim resource service losses and the methods for quantifying them, and the logic for selecting planting sites has given us an unprecedented ability to foster effective restoration of these habitats. More importantly, perhaps, is the signal that this has sent to the development community and responsible parties: that this resource is of vital national importance and its destruction cannot be tolerated by the public. While transplanting seagrass is not technically complex, in order to meet the goal of maintaining or increasing seagrass area, careful attention to detail must be paid to the entire process of planning, planting and monitoring – a process that does not lend itself to oversimplification. As with all terrestrial crops, there are inherent risks with seagrass and failures will inevitably occur. Given that despite collective millennia of human experience we trade stock futures on the probability of successful cultivation of food crops, restoration of seagrass ecosystems will suffer from at least this kind of risk.

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